Ecological Consequences of PCBs in the Exposed Sediments of Formerly Impounded Areas of the Kalamazoo River

Overview of Studies Conducted by Michigan State University

Prepared on behalf of the Kalamazoo River Study Group and the USEPA by:

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1 Introduction

The Kalamazoo River is located in southwestern Michigan, and flows in a northwesterly direction, through Kalamazoo and Allegan Counties to Lake Michigan. It drains approximately 2,000 square miles and is fed by more than 400 miles of tributaries. The Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site (Site), includes 80 miles of the Kalamazoo River from Morrow Lake Dam downstream to Lake Michigan, including the river banks and formerly impounded floodplains, as well as a 3-mile stretch of Portage Creek and four areas used as disposal sites for paper-making residuals. The contaminants of concern for the Site are polychlorinated biphenyls (PCBs), which have been found in sediment and surface water, floodplain soils, and aquatic and terrestrial biota.

An initial Baseline Ecological Risk Assessment (Baseline ERA), prepared on behalf of the Michigan Department of Environmental Quality (MDEQ), was released in June of 1999 (Camp Dresser and McKee [CDM], 1999). Subsequent to the release of the initial Baseline ERA, the Kalamazoo River Study Group[1] (KRSG) began funding a series of studies by Michigan State University (MSU) to develop Site-specific information on exposure of representative receptor species to PCBs and the potential ecological risk posed by that exposure. Professor John P. Giesy, then a Distinguished Professor of Zoology at MSU was the principal investigator for the MSU studies and Dr. Matthew Zwiernik was the project leader. The research was conducted through the auspices of the National Food Safety and Toxicology Center (NFSTC) and the Center for Integrative Toxicology (CIT) at MSU. Currently, Dr. Giesy is Distinguished Professor Emeritus of Zoology at MSU and Professor and Canada Research Chair in Environmental Toxicology at the University of Saskatchewan, Saskatoon, Saskatchewan, Canada. Professor Zwiernik is currently a Professor at the NFSTC and Adjunct Professor of Zoology and of Animal Science at MSU.

We designed our studies to provide Site-specific information on exposure and effects of PCBs on selected wildlife species. The information was collected with the primary goal of reducing uncertainty in the Baseline ERA (CDM, 1999) by reducing the reliance on assumptions in lieu of site observations. Our intent was to develop multiple, Site-specific, independent lines of evidence to supplement those evaluated in the Baseline ERA (CDM, 1999). As an example, we determined dietary exposure for several avian species based on site-specific observations of dietary composition and measurement of PCB concentrations in dietary items for the species being evaluated. In addition, we evaluated exposure directly by measuring the PCB concentrations in critical tissues (e.g., eggs) of the same ecological receptors. Hazard assessments for both study and reference

[1] Georgia-Pacific Corporation and Millennium Holdings, LLC are collectively referred to as the Kalamazoo River Study Group, or KRSG

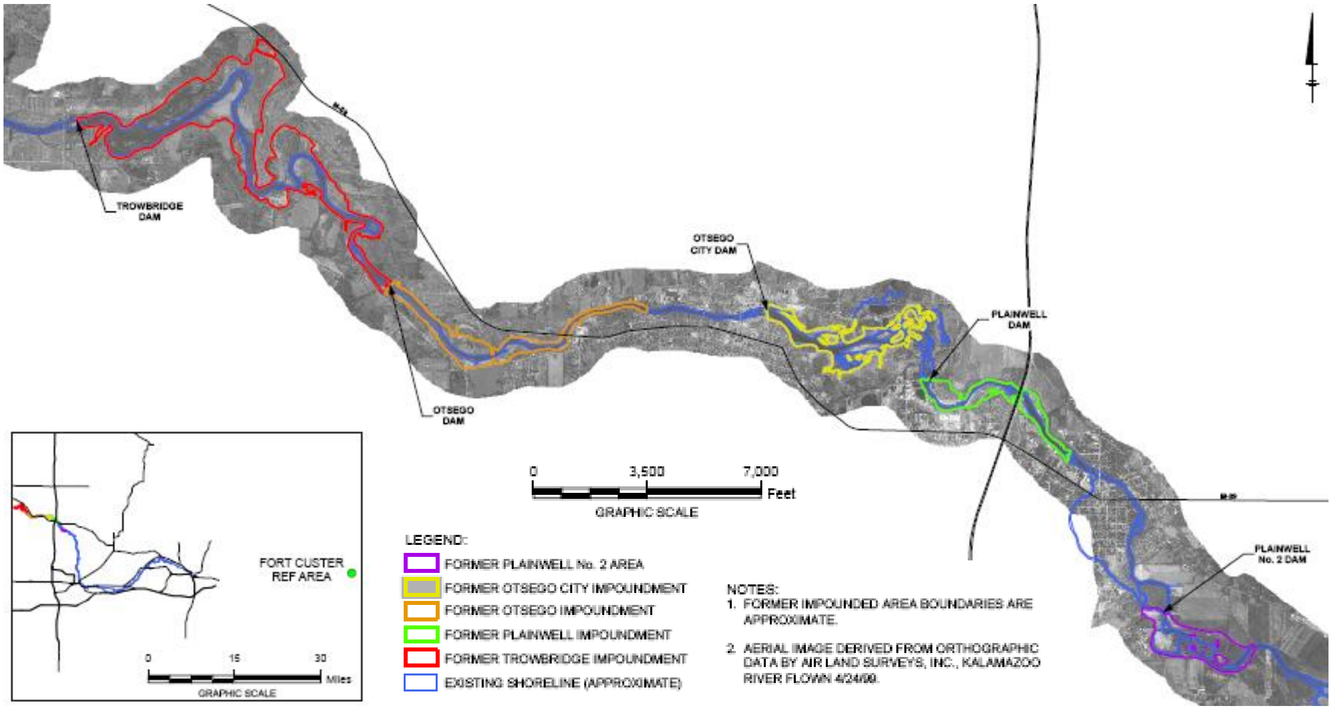
areas were then made for these species based on the two distinct estimates of exposure and compared to Site-specific measures of effects including fecundity, growth, and productivity.

Our studies were conducted largely in the floodplains of the former Trowbridge Impoundment. There are five areas of the Site – the historically inundated area around the Plainwell No. 2 Dam and the formerly impounded areas associated with the dams at Plainwell, Otsego City, Otsego, and Trowbridge – where the current floodplains include historically submerged sediments that were exposed when the dams were opened, dismantled, or taken down to the sills. These exposed sediments, which had accumulated PCBs while submerged, became the soils of the newly exposed floodplains. The history of these impoundments, summaries of the PCB concentrations and distributions, habitat characterization, and descriptions of wildlife are presented in the report *Characteristics of the Formerly Impounded Areas* (ARCADIS 2008; included as Attachment 1). We chose the former Trowbridge Impoundment because:

1) the available habitat types, presence of receptor species, and mixture of PCB congeners found there are generally consistent with those of the other former impoundments;
2) PCB concentrations in exposed sediments were comparable to those of the Otsego and Plainwell Impoundments; and,
3) it is the largest of the formerly impounded areas and consequently provides the opportunity to study populations of receptors whose foraging is most concentrated in habitats developed on exposed sediments.

In addition, we established a reference site in the Fort Custer State Recreational Area, which is some 40 kilometers (km) upstream of the former Trowbridge Impoundment and well upstream of any Site-related sources of PCBs. We selected the Fort Custer Recreation Area as a reference site because of previously identified low or background concentrations of PCBs as well as other contaminants of concern (COCs) based on previously collected data (CDM, 1993 and 1997; BBL, 1993, 1994a, b, c; Giesy et al., 1994a). The reference site provided similar, but not identical, floodplain habitat with minimal exposure to PCBs. Parallel studies were conducted in the former Trowbridge Impoundment and the reference site to distinguish potential Site-related effects. The locations of the former Trowbridge Impoundment, the Fort Custer reference site, and the four other formerly impounded areas are shown on Figure 1-1.

Figure 1-1. Formerly Impounded Areas of the Kalamazoo River and the Fort Custer Reference Area



Although the studies were conducted to gather data that could be used to evaluate potential risks associated with PCB exposure in both aquatic (Millsap et al., 2004; Kay et al., 2005; Neigh et al., 2006c and d) and terrestrial food webs, the focus of this document is to provide a summary of only those efforts related to analyses of potential risks to terrestrial receptors who might be exposed to the exposed sediments of the floodplain habitat in the formerly impounded/historically inundated areas.

The results of our studies for terrestrial floodplains have been described in eight publications in the peer-reviewed literature (Attachment 3) and have provided the basis for two Ph.D. dissertations. These results and all of the underlying data have been provided to the United States Environmental Protection Agency (USEPA) and MDEQ for review and consideration. However, USEPA and MDEQ were not involved in the design or implementation of these studies, and the Final Revised Baseline ERA (CDM, 2003; Attachment 2) was completed before our studies were finalized. Therefore, the KRSG and USEPA have agreed to conduct a peer review of our ecological studies performed to date with respect to floodplain soils, prior to consideration of the results of these studies as independent lines of evidence in subsequent Area-specific ecological risk assessments to be conducted in the floodplains of formerly impounded areas of the Kalamazoo River.

This document summarizes our studies evaluating risk to populations exposed to PCBs in soils of the formerly impounded areas. These include two passerine species, (eastern bluebird [*Sialia sialis*] and house wren [*Troglogytes aedon*]), (Neigh et al., 2006a, b, and 2007), great-horned owl (*Bubo virginianus*) (Zwiernik et al., 2007; Strause et al., 2007a, b and 2008), and the short-tailed shrew (*Blarina brevicauda*). Full details of the methods applied and the results observed can be found in the specific studies referenced, which are included as Attachment 3.

1.1 Overview of MSU Studies

We performed a series of comprehensive field studies to evaluate potential risk to representative ecological receptors posed by elevated concentrations of PCBs in soils in the formerly impounded areas of the Kalamazoo River. These studies, which are included in Attachment 3, were intended to support risk management decisions regarding these soils. As recognized by the USEPA (USEPA, 1994), site-specific field studies are important for sound decision making when complex systems are involved. This is due to the level of uncertainty involved in predicting wildlife contaminant exposure and the potential for effects in complex systems. Data were collected to address; spatial and temporal considerations; receptor priority; and, utility as exposure- and effect-measurement endpoints.

Prior to our studies, substantial amounts of data on the concentrations and spatial distributions of PCBs in floodplain soils of the formerly impounded areas were available (BBL 1994a and b). However, we observed uncertainties that could not be resolved with the existing data sets and designed Site-specific studies to address these uncertainties and to provide multiple lines of evidence for evaluation of risk. Each line of evidence was designed as a key measurement endpoint that could be directly associated with assessment endpoints established in the Baseline ERA. Although, the research was conducted as a student-based research program, we administered a rigorous quality assurance (QA) program to assure usability of the studies for both scientific publication and Superfund risk assessment.

As described above, our studies were designed to provide Site-specific data, encompassing multiple years and locations for the purpose of facilitating a weighted multiple line-of-evidence approach (Fairbrother, 2003; Hull et al., 2006). Studies included productivity measures for the two passerine species and the great horned owl, as well as measures of Site-specific dietary and tissue-based exposure. Site-specific dietary exposure was assessed by determining the Site-specific dietary composition of selected receptor species and then sampling those dietary items from where they were being consumed and analyzing them for PCBs. These data were then used to reconstruct the Site-specific daily dietary exposure for each receptor for comparison to a dietary toxicity reference value (TRV). This comparison resulted in a numerical hazard quotient (HQ) which was used in conjunction with other lines of evidence to evaluate potential risk to the specified receptor. Likewise, the tissue-based assessment was evaluated based on direct measurements of PCBs in tissues compared to tissue-based TRVs. Dietary and tissue-based assessments were done on both total PCBs (ΣPCBs) and toxicity equivalency quotients (TEQ) basis for avian species (Neigh et al., 2006 a, b; Strause et al., 2008). Concentrations of TEQ in bird tissues were calculated by summing the product of the concentrations of individual PCB congeners and their respective, bird-specific World Health Organization (WHO) toxic equivalency factors (TEF) (Van den Berg et al., 1998). The results of these assessments are summarized herein and a more detailed discussion of each assessment can be found in the publications (Neigh et al., 2006a, b; Strause et al., 2008) which are included as Attachment 3.

Individual receptor species were prioritized as to importance based on intensity of exposure, relative sensitivity to PCBs, prevalence of preferred habitat within the Kalamazoo River areas of concern (AOC), and appropriateness as a surrogate species, with special considerations given to those species previously identified (1999 initial and 2003 Final Revised Baseline ERA) as receptors of concern (see Table 1-1, below).

Table 1-1. Terrestrial Receptors Considered in MDEQ's Baseline ERA and MSU Site-Specific Studies

Comprehensive Receptor List	Feeding Guild	Baseline ERA (CDM 2003)	MSU
Omnivorous/Insectivorous Birds			
House Wren	Terrestrial insectivore (passerine)		√
Bluebirds	Terrestrial omnivore (passerine)		√
American Robin	Terrestrial omnivore (passerine)	√[1]	√
Carnivorous Birds			
Great Horned Owl	Terrestrial avian carnivore (raptor)	√	√
Small Insectivorous/Omnivorous Mammals			
Short-Tailed Shrew	Terrestrial carnivore/insectivore (mammal)		√
Deer Mouse	Terrestrial omnivore/herbivore (mammal)	√[2]	
Carnivorous Mammals			
Red Fox	Terrestrial omnivore	√[2]	

Notes:
1. Receptor evaluated as a secondary representative species for receptor class
2. MDEQ Baseline ERA found no significant risk.

Assessments of exposure through the diet focused on six biological sampling areas (BSAs), each proximal to the Kalamazoo River, associated with similar habitats, entirely within the 100-year floodplain, and all on property owned by the state of Michigan. The two reference site BSAs were located within the Fort Custer Recreation Area. The remaining four BSAs were located in the former Trowbridge Impoundment.

Terrestrial dietary items evaluated in our studies were collected from or in the vicinity of these BSAs to provide a site-specific measure of actual dietary exposures. Tissue-based exposure methodologies were used to directly quantify embryo (egg) and developmental (nestling) exposures, confirm dietary exposures, and address uncertainties associated with foraging range and residence time. Tissues analyzed included those of adults (birds and mammals), juveniles (birds and mammals), nestlings (e.g. passerine chicks, raptor plasma), and eggs (all avian receptors). Small mammals for analysis were collected entirely from within the BSAs, passerine tissues were collected from within or proximal to BSAs, and raptor tissues were collected within, proximal, and outside the BSAs. The sample locations for all data collected by MSU from the Site study area (i.e., the former Trowbridge Impoundment) are shown in Figure 1-2 (figure to be provided when complete).

1.2 Quality Assurance and Quality Control Procedures

A quality assurance program was developed and strictly applied to assure that the methods used were consistent with the available guidance and state of the science so that the data would be of appropriate quality to support risk assessment and risk management activities. A Sampling and Analysis Plan (SAP) (Attachment 4) was developed to govern our studies. In addition, before any work was conducted detailed standard operating procedures (SOPs) (see Attachment 4 for individual SOPs) were developed for all phases of the research. The details of the methods, and in particular the quality assurance guidance, are described in those documents. The SAP, supporting SOPS, and the Quality Assurance Project Plan (QAPP) were written based on USEPA guidance for studies at Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) sites. The results have been subject to extensive internal and external peer review through the processes outlined in the QAPP (Attachment 5). In addition, all data were subject to the highest level quality assurance/quality control (QA/QC) evaluation. This was accomplished through oversight from ENTRIX Inc, which provided QA planning, conducted QA audits, and archived samples and data such that it could be applied to decision-making. The database for our studies, as well as all of the raw data including the QA reports for all of the analytical results have been provided to EPA and the State and are available upon request. These reports include all of the chromatograms, calculations, data reductions, and QA narrative reports, as well as data summaries. In addition, extensive pictorial documentation of locations and samples (including pictures of the sample, date and time, and global positioning system [GPS] location), field notes (scanned from field notebooks), field record sheets, subsequent sample tracking sheets, and chain of custody records are available. For instrumental analyses, there are electronic and hard copy archives of all sample tracking sheets used to track the progress of a sample from the field, to archive, and through the analytical procedures. This additional information can be provided upon request. The SOPs (Attachment 4) describe how these procedures and records were used.

1.3 Report Overview

Section 1: Introduction

Section 2: Assessment of Potential Risk to Terrestrial Omnivorous/Insectivorous Passerine Birds (Eastern Bluebird and House Wren)

Section 3: Assessment of Potential Risk to Carnivorous Terrestrial Birds (Great Horned Owl)

Section 4: Assessment of Potential Risk to Small Terrestrial Mammals (Short-Tailed Shrew)

Section 5: Conclusions

Section 6: References

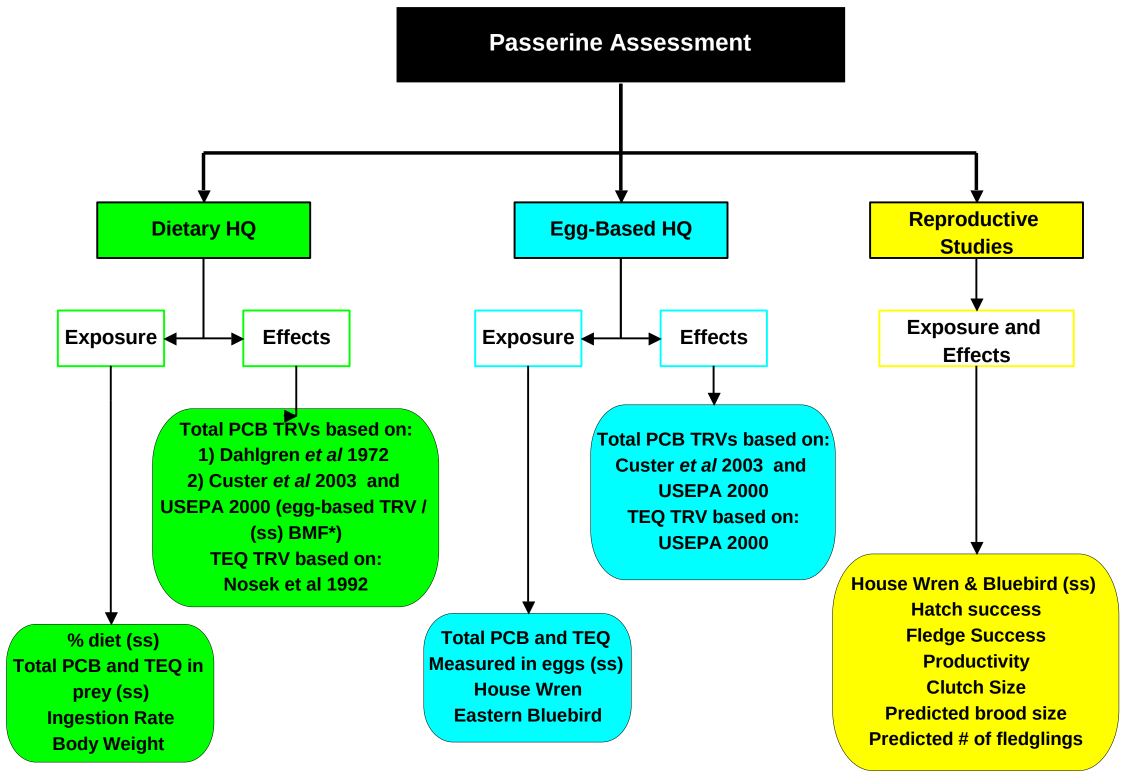
2 Assessment of Potential Risk to Terrestrial Omnivorous/Insectivorous Passerines (Eastern Bluebird and House Wren)

In the CDM Final Revised Baseline ERA (2003), the American robin was identified as one of the representative receptors for evaluation of terrestrial avian species, specifically for the omnivorous feeding guild. The Final Revised Baseline ERA (CDM, 2003) found that "omnivorous birds that consume substantial amounts of soil invertebrates (e.g., earthworms) would be at significant risk if foraging takes place in mostly contaminated areas." This conclusion was based primarily on a dietary HQ analysis that compared Site-wide measures of PCBs concentrations in plant and earthworm tissues to literature-derived TRVs based on domestic chickens.

While the results presented in the Final Revised Baseline ERA (CDM, 2003) were not complete at the time that our studies were undertaken, preliminary results presented in the Final Draft ERA (CDM, 1996) and in the initial draft of the Final Baseline ERA (CDM, 1999) were available and indicated that omnivorous birds feeding in the floodplain of the Kalamazoo River AOC would potentially be at risk from exposure to PCBs. The passerine studies were undertaken to provide additional Site-specific information regarding potential risk to omnivorous birds feeding in the floodplain to support risk management decision making. The available lines of evidence for this receptor group include direct measures of reproductive output of the eastern bluebird and house wren as well as HQs based on both dietary and tissue-based exposure (Neigh et al., 2006a and b, and 2007).

The lines of evidence and associated inputs evaluated in our assessment are summarized in Figure 2-1. Based on a number of species characteristics including dietary composition, foraging characteristics, Site use, exposure potential, and ease of study, the house wren was selected as an example of an insectivore. The eastern bluebird was selected as the example omnivorous terrestrial passerine based on exposure potential, perceived sensitivity, and societal value. Both of these species are amenable to nesting in provided nest boxes, which allowed control of the feeding environment and facilitated collection of samples of adults, eggs, chicks, and food samples (boluses) from which dietary composition and PCBs content could be determined.

Figure 2-1. Lines of Evidence Evaluated by MSU for Passerine Species



(ss) = site specific

* BMF = site- and species-specific biomagnification factor to estimate a dietary exposure from an egg concentration

2.1 Measures of Reproductive Success

Studies of reproductive performance of the house wren and eastern bluebird were conducted over the course of three consecutive years (Neigh et al., 2007). A total of seventeen bluebird and 34 house wren nests were established at the Site study area (i.e., the former Trowbridge Impoundment) over the course of the study period while 57 and 71 bluebird and wren nests respectively were completed at the reference site (Fort Custer). Nest boxes were monitored for clutch initiation, clutch size, hatch day, hatching success (percentage of eggs hatched), fledging success (percentage of nestlings fledged per nestling hatched), and productivity (percentage of nestlings fledged per egg laid). Additional comparisons included nest attentiveness and temporal comparisons of nestling growth.

House wrens made a greater number of nesting attempts than eastern bluebirds at both the Site and reference study areas. This difference was expected based on observations of species abundance and available habitat. Abandonment rates for both species ranged between 0% and 25%, which was similar to the range reported for unexposed bluebirds elsewhere (Rustad, 1972). Reproductive parameters for house wrens at each location were not significantly different between years, therefore all years were combined. Over the duration of the study there were no differences between the Site study area and reference site for clutch size, hatching success, or overall productivity. The only statistically significant difference noted in house wren reproduction was a significant increase in fledging success for nests in the Site study area. Clutch size, hatching success, and fledging success were also not different between the Site and reference study areas for bluebirds over the duration of the study. However, eastern bluebird productivity was significantly less in the Site study area ($p<0.05$, Mann-Whitney U). This was largely based on a single female that twice attempted to produce a brood in 2001 without success but did not abandon the eggs. This one female accounted for a 40% decrease in productivity during that year and if removed from the data set, the difference is not significant. Even including this individual the ranges of productivity (as identified by the mean and standard deviations) for the Site and reference study areas overlap. The results of the productivity analysis are summarized below in Table 2-1.

Other observational data for these studies included measures of individual condition. There were no gross physical abnormalities observed in adults ($n=366$) or nestlings ($n=542$) for any of the passerine species monitored during the study. There were no differences between the Site and reference study areas in adult nest attentiveness or in any of the nestling growth parameters for either species over the duration of the study. Moreover, there were no embryo abnormalities noted in any of the 305 eastern bluebird eggs and 607 house wren eggs examined. Additionally, there were no differences in egg mass for either species or in egg length in house wrens. Egg lengths for the eastern bluebird were significantly greater at the reference site ($P<0.05$ Students t test); however, the difference in mean length was less than 1 millimeter (mm) which was less than 8% of the total length, and the egg length range of means plus one standard deviation overlapped. There were abnormalities of eggs noted in house wren eggs at both the Site and reference areas including irregular shape, abnormal texture, and desiccated contents.

As discussed previously, measures of productivity were not significantly different between the Site and reference study areas for house wrens. Productivity for bluebirds was slightly less at the Site study area than at the reference area, largely due to small sample size at the Site study area and the resulting impact of the single female in 2001 that twice attempted to produce a brood without success but did not abandon the eggs. Measures of the condition of individuals were normal with the exception of the house wren egg abnormalities mentioned above. Given that no abnormalities were noted in any embryos or nestlings and that nestling growth was not affected, these egg abnormalities do not appear to have had an adverse impact on productivity.

Table 2-1. Nest Productivity Measurements for Eastern Bluebirds and House Wrens at the Former Trowbridge Impoundment and the Ft. Custer Reference Area

	Trowbridge[1]		Ft. Custer[1]	
	n	**Mean**	**n**	**Mean**
Eastern Bluebird				
Hatching success[2]	14	59	49	0.79
Fledging success[3]	9	0.83	45	0.96
Productivity[4]	13	0.47*	49	0.76
Clutch size	18	3.6	57	4.2
Predicted brood size	10	3.3	45	3.8
Predicted number of fledglings	8	3	43	3.9
House Wren				
Hatching success[2]	31	0.64	60	0.81
Fledging success[3]	25	1.0*	59	0.92
Productivity[4]	31	0.64	60	0.74
Clutch size	36	5.4	71	5.7
Predicted brood size	25	4.6	59	5
Predicted number of fledglings	25	4.6	56	4.8

[1] Sample size (n) and mean values are for all clutches. For information on early and late clutches, see Table 3 in Neigh et al. 2007

[2] Hatching success is calculated as the number of eggs hatched per egg laid

[3] Fledging success is calculated as the number of fledglings per nestling hatched

[4] Productivity is calculated as the number of fledglings per egg laid

* Mean of there Trowbridge population is statistically different from the Ft. Custer population ($P<0.05$)

2.2 Hazard Quotient Analyses

In addition to the direct measures of productivity discussed above, terrestrial insectivorous and omnivorous birds were assessed based on an HQ analyses as additional lines of evidence. The HQ analyses included the comparison of estimates of exposure to appropriate TRVs. Each of these elements and the resulting HQs are discussed in the following sections. Exposures and risks were calculated using both ΣPCBs and TEQ approaches to estimate exposures.

2.2.1 Exposure Calculations

Exposure of passerine birds to ΣPCBs was assessed in terms of daily dose for dietary exposure and direct measures of tissue concentrations in adults, nestlings, and eggs for tissue-based exposure. Corresponding toxicity reference values are not available for adult and nestling tissues; however, such values are available for dietary exposure and eggs. Consequently, the exposure estimates described below focus on dietary and egg-based exposure. Details of other tissue measurements can be found in Neigh et al. (2006a and b).

2.2.1.1 Dietary Exposure Estimates

Dietary exposure for the house wren and eastern bluebird was estimated by calculating an average potential daily dose (APDD) for each species (Table 2-2). The dose model used is consistent with the model recommended in the USEPA's Wildlife Exposure Factors Handbook (USEPA, 1993) and is described in more detail in Neigh et al. (2006a). Site-specific inputs to the APDD for each receptor species include Site-specific dietary composition of each target species (Figure 2-2) and concentrations of ΣPCBs and TEQs in each identified dietary item (Table 2-2). Composition of the diet of each passerine was determined by quantifying stomach contents and bolus compositions. Identified prey items were collected from the locations where they were being consumed (i.e., from six individual BSAs that were co-located with passerine study areas) on sampling dates that roughly corresponded to spring, early summer, and late summer in 2000 and 2001. The nest box locations and prey tissue sample locations are provided in Figure 1-2. Additional details of the daily dose calculation and inputs can be found in Neigh et al. (2006a). It should be noted that although these studies focused on house wrens and bluebirds as representatives of select feeding guilds, the Site-specific dietary item sampling data could be used to estimate dietary exposures for many potential receptors. The representativeness of these two species is discussed in more detail in Section 2.4.

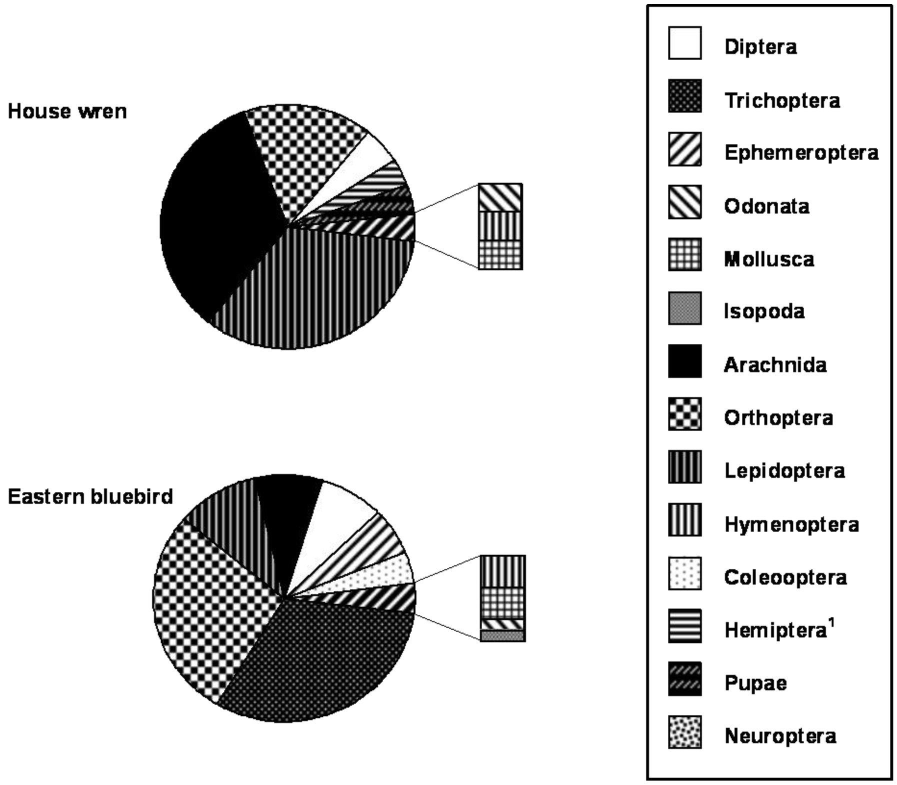
Table 2-2. ΣPCB and TEQ Concentrations in Biota and Average Potential Daily Dose for Eastern Bluebirds and House Wrens From the Former Trowbridge Impoundment

	Number of Samples	Mean ΣPCBs (± std deviation)	Number of Samples	Mean Avian TEQs (± std deviation)
Biota Concentrations		**(mg/kg)[1]**		**(ng/kg)[1]**
Plants	28	0.023 ± 0.044	--	--
Terrestrial Invertebrates[2]	30	0.34 ± 0.57	30	80 ± 200
Fresh Earthworms	18	1.72 ± 1.75	18	238 ± 255
Depurated Earthworms	14	1.26 ± 1.13	14	154 ± 186
Average Potential Daily Dose		**(mg/kg/day)**		**(ng/kg/day)**
Eastern Bluebird APDD		0.51		70
House Wren APDD		0.13		31

Notes:

1. PCBs concentrations are wet weight basis for biota
2. Earthworms were treated separately from other terrestrial invertebrates

Figure 2-2. Dietary Composition of Passerine Species Evaluated



Notes:
[1]The order Hemiptera also includes the order Homoptera.

2.2.1.2 Tissue Based Exposure Estimates

Samples of eggs were collected for bluebirds and house wrens during the spring and summer of 2001, 2002, and 2003. A maximum of one tissue sample was collected randomly from each nest box for quantification of ΣPCBs and TEQs concentrations. In addition, abandoned and addled eggs were taken for additional measurements of ΣPCB and TEQ concentrations. Eggs were sampled 7 to 10 days after they were laid (Table 2-3). Table 2-3 also includes the concentrations of ΣPCBs in adult tissues. As previously discussed, no TRVs are available for comparison to concentrations in adults, therefore exposure estimates based on adult tissue were used only to assess relative exposure among the study species and other passerine species that use the Kalamazoo River floodplain (i.e., the American robin and tree swallow). The tree swallow had the greatest concentrations of ΣPCBs, followed by the house wren, and then the robin. This is likely a result of a greater proportion of the tree swallow's diet consisting of aquatic insects which were found to have some of the greatest PCBs concentrations for insects measured (Kay et al., 2005).

Table 2-3. Mean Concentrations of ΣPCBs and TEQs in Passerine Tissues in the Former Trowbridge Impoundment

	Number of Samples	PCBs (mg/kg)	Number of Samples	Avian TEQ (ng/kg)
Eastern bluebird[1]				
Egg	7	8.3 (5.1)	5	77 (82)
House wren[1]				
Egg	14	5.8 (4.4)	11	360 (330)
Adult	9	3.2 (2.1)	9	110 (57)
Tree Swallows[2]				
Adult	9	8.7(9.7)	9	0.29 (0.23)
American Robin[3]				
Adults	17	0.92 (1.2)	17	3.9 (4.4)

Notes:
( ) - standard deviation
[1] Mean values taken from Neigh et al., 2006b
[2] Mean values taken from Neigh et al., 2006c
[3] Mean values taken from Blankenship et al., 2005

2.2.2 Effect Levels (i.e., Toxicity Reference Values)

TRVs were derived from information in the literature according to USEPA guidance. TRVs discussed herein were based on concentrations of ΣPCBs and TEQs in the diet and eggs. The selection of TRVs was based on several criteria to determine their appropriateness for use in this study. These criteria included: 1) the use of wildlife species rather than a laboratory species where possible; 2) chronic exposure and sensitive life stages; 3) the evaluation of ecologically relevant endpoints; 4) minimal co-contamination; 5) multi-year studies; 6) and ΣPCBs concentrations were reported or could be calculated. There were few studies available for passerine species and there were no studies with a similar passerine exposure profile. Thus, a range of TRVs were developed to address possible variability in species-sensitivity and test conditions. It has been established that species-sensitivity to aryl hydrocarbon receptor (AhR) mediated toxicity is directly related to the binding affinity of its Ah receptors for the compound of interest (in this case PCBs) (Karchner et al., 2006). As such, it is important to recognize the importance of selecting appropriate TRVs as well as considering multiple lines of evidence to understand the potential for adverse effects. The different sensitivities of various test species are discussed in more detail in Section 2.4. The selection of TRVs for evaluation of terrestrial feeding passerines in the Kalamazoo River floodplain is described in the following two sections and the selected values are summarized in Table 2-4, below.

2.2.2.1 Dietary TRVs

The dietary TRVs for ΣPCBs based on the no observed adverse effect level (NOAEL) or lowest observe adverse effect level (LOAEL) selected for this evaluation, as well as for USEPA applications (USEPA, 1995; 2000), were derived from a study in which ring-necked pheasants (*Phasianus colchicus*) were dosed with Aroclor 1254 and a threshold for effects was assessed for critical reproductive life stages (Dahlgren et al., 1972). The rationale for the TRV selection is described in Neigh, et al. (2006a). Wild passerine birds, such as those examined in this study, appear to be less sensitive to PCBs exposure (Thiel et al., 1997; Custer et al., 1998; Henning et al., 2003) than domesticated *Galliformes* on which the dietary TRVs are based (e.g. ring-necked pheasant; Dahlgren et al., 1972). To address this uncertainty, an alternative set of dietary TRVs were developed using studies conducted on wild passerine species exposed to PCBs (Custer et al., 2003; USEPA, 2000). Because these studies only provided egg tissue concentrations that can be associated with either no or few adverse effects (i.e., they did not estimate dietary exposure), a dietary TRV was extrapolated from the egg-based TRVs using Site- and species-specific biomagnification factors (BMFs). BMFs are calculated by dividing measured egg concentrations by Site- and species-specific dietary dose estimates. The diet and egg inputs and resulting BMFs and TRVs for the house wren and bluebird are summarized (Table 2-4). The BMF calculation is described in more detail in Neigh et al. (2006a). The egg-based TRVs used in the BMF approach are described in Section 2.2.2.2.

No studies of the effects of TEQs in passerines were found in the literature. In the absence of such data, a dietary TEQ-based LOAEL of 140 ng TEQ/g bw/d was derived (Neigh et al., 2006a) based on a sub-chronic laboratory study evaluating the effects of intra-peritoneal injections of 2,3,7,8-TCDD at concentrations of 1000 ng TCDD/g/week on ring-necked pheasants (Nosek et al., 1992). A NOAEL (14 ng TEQ/g bw/d) was derived as well by applying a safety factor of 10 (Strause et al., 2008). TEQ based TRVs were also derived from Site-specific BMFs as described above and are presented in Table 2-4.

2.2.2.2 Egg-based TRVs

In addition to a dietary assessment, risk was assessed through HQs by comparing concentrations of ΣPCBs measured in eggs to tissue-based TRVs based on the NOAEL and the LOAEL for ΣPCBs. Concentrations of PCBs in eggs are the most direct and ecologically relevant measurement endpoint for predicting adverse effects on reproduction induced by ΣPCBs because they provide an empirical measure of exposure (i.e., the tissue burden) rather than a modeled exposure (i.e., dietary exposure), which even with Site-specific inputs still relies on a number of generic assumptions (i.e., assumed ingestion rate, absorption rate, etc). HQs calculated from the exposure of eggs to ΣPCBs were considered to be a maximum estimate of exposure to a sensitive life stage (Giesy et al., 1994a).

Although appropriate toxicity data were not available for an exclusively terrestrial, avian species, TRVs were available for the tree swallow, a wild passerine species. The egg-based TRVs selected for this assessment were derived from several studies conducted using tree swallows on the Hudson River and Housatonic River. On the Hudson River, potential effects in tree swallows were monitored over the course of two years. Effects such as abnormal plumage, decreased hatching success, and increased abandonment were observed in one year but not the other. The NOAEL value of 26.7 mg ΣPCBs/ kg wet weight (ww) represents the lowest Site-wide mean concentration of PCBs in eggs from any of the locations evaluated during the year in which no effects were observed. The LOAEL for ΣPCBs was derived from a Housatonic River study in which hatching success was impaired during two years of the study (Custer et al., 2003). The lowest average concentration of ΣPCBs in piping chicks from the two years (63 mg ΣPCBs/kg) was used as the LOAEL. This study was deemed appropriate because effects were observed in two consecutive years and sensitive reproductive endpoints were evaluated.

The TRV selected as a threshold for effects based on egg TEQ concentrations was also based on the Hudson River study (USEPA 2000). The greatest TEQ concentration in the year without effects, 13,000 ng TEQ/kg ww was selected as the NOAEL value. A LOAEL based on TEQs could not be derived from the literature (Neigh et al., 2006a and b).

Table 2-4. Toxicity Reference Values for Dietary and Egg-Based Exposure of Avian Species to ΣPCBs and TEQs

Reference	BMF	TRV
ΣPCBs		
Dietary TRVs (mg/kg/d)		
Dahlgren *et al*. (1972)	--	LOAEL = 1.8 mg/kg/d
		NOAEL= 0.6 mg/kg/d
	Bluebird BMF (16) = mean egg ΣPCBs concentration (8.3 mg/kg ww) / dietary exposure estimate (0.51 mg/kg/day)	LOAEL = 3.9 mg/kg/d[1]
		NOAEL = 1.6 mg/kg ww[2]
	House Wren BMF (45) = mean egg ΣPCBs concentration (5.8 mg/kg ww) / dietary exposure estimate (0.13 mg/kg/day)	LOAEL = 1.4 mg/kg/d[1]
		NOAEL = 0.6 mg/kg/d[2]
Egg-based TRVs (mg/kg wet weight)		
Custer et al. (2003)	--	LOAEL = 63 mg/kg wet weight
USEPA (2000)	--	NOAEL = 26.7 mg/kg wet weight
TEQs		
Dietary TRVs (mg/kg/d)		
Nosek et al. (1992)	--	LOAEL = 140 ng TEQ/kg/d
		NOAEL= 14 ng TEQ/kg/d
	Bluebird BMF (1.1) = mean egg TEQ concentration (77 ng/kg) / dietary exposure estimate (70 ng/kg/day)	LOAEL = NA
		NOAEL = 11,818 ng TEQ/kg/d[1]
	House Wren BMF (11.6) = mean egg TEQ concentration (360 ng/kg ww) / dietary exposure estimate (31 ng/kg/day)	LOAEL = NA
		NOAEL = 1119[2]
Egg-based TRVs (ng TEQ/kg ww)		
	--	LOAEL = NA
USEPA (2000)	--	NOAEL = 13,000 ng/kg ww

Notes:

1 Calculated from TRV selected from Custer et al. (2003) / species BMF.

2 Calculated from TRV selected from USEPA (2000) / species BMF.

NA Value not available

-- BMF not applicable

2.2.3 Results

Both dietary and egg-based HQs for ΣPCBs and TEQs are described below. These and other results are described in more detail in Neigh et al. (2006a and b).

2.2.3.1 Dietary HQs

HQs based on dietary ΣPCBs and mean APDD in the Site study area were less than 1.0 for both NOAEL and LOAEL TRVs for both passerine species based on Site-specific dietary items and Site-specific dietary composition for house wrens and bluebirds. The HQs for the dietary TEQs and mean APDD were higher with the NOAEL HQs greater than 1 (i.e., 5.0 and 2.2 for the bluebird and wren respectively) and LOAEL HQs less than one for both species (Table 2-5). Species were also evaluated at the upper 95% upper confidence limit on the mean (U95 CL) for APDD in order to describe a range of ΣPCB and TEQ HQs that would represent the most sensitive portion of the population. NOAEL U95 CL HQ values for the bluebird were greater than one but LOAEL HQs were less than one. Both NOAEL and LOAEL U95 CL HQs for the house wren were less than one. The dietary HQs based on the BMF approach were all less than one (i.e., for both species, for ΣPCBs and TEQs and for NOAEL and LOAEL TRVs). The mean and U95 CL concentrations for each species and the Site-specific diet are presented graphically (Figure 5 in Neigh et al., (2006a). Additional details of the analyses described here can also be found in the publication (Neigh et al, 2006a).

2.2.3.2 Tissue Based HQs

HQs based on mean and U95 CL ΣPCB and TEQ concentrations for the Site study area were well below 1 for both species. Moreover, no sample (i.e., nest box) exceeded the NOAEL egg-based TRV of 26.7 mg/kg (i.e., the maximum detected ΣPCB concentration in bluebird eggs was 15.6 mg/kg and in house wren eggs was 8.3 mg/kg). Mean HQ values are summarized Table 2-5.

Table 2-5. Summary of Hazard Quotients for Terrestrial Passerines in the Former Trowbridge Impoundment

	Mean ΣPCB HQs		Mean TEQ HQs	
Receptor	**NOAEL**	**LOAEL**	**NOAEL**	**LOAEL**
Dietary				
Bluebird	0.85	0.28	5.0	0.50
Wren	0.22	0.072	2.2	0.22
Dietary using BMF				
Bluebird	0.31	0.13	0.006	NA
Wren	0.22	0.092	0.028	NA
Egg-Based				
Bluebird	0.31	0.13	0.0059	NA
Wren	0.22	0.092	0.028	NA

Notes: Toxicity Reference Values used HQ calculations can be found in Table 2-4
NA = no toxicity reference value available. HQ could not be calculated.

2.3 Assessment of Terrestrial Passerine Risk Based on Multiple Lines of Evidence

A multiple lines of evidence approach for an ERA reduces uncertainty and provides the most accurate evaluation of potential hazard (Fairbrother, 2003; Hull et al., 2006). The approach includes various types of data and integrates the strengths and weaknesses of each into a conclusion that best describes the entire data set.

Based on an extensive multi-year, Site-specific exposure and effects assessment, there does not appear to be significant risk of adverse effects for terrestrial passerine birds exposed to PCBs (either as ΣPCBs or TEQs) originating from floodplain soils of the Kalamazoo River. This conclusion is based on data which suggests there is neither a significant implication for cause, nor an identification of effects consistent with ecologically relevant risk.

Dietary and tissue-based exposure assessments demonstrated that terrestrial passerines in the former Trowbridge Impoundment are accumulating increased concentrations of ΣPCBs relative to the reference study area (Neigh et al., 2006b). However, multiple lines of evidence, taken together, indicate that this increased exposure to ΣPCBs in floodplain soils is not associated with increased risk to these species. Egg-based HQs were all less than 1.0 as were the majority of the dietary based HQs. The TEQ analyses tended to result in greater (passerines) or similar (great-horned owl) HQs than the ΣPCBs analyses. The TEQ calculations are considered less certain than the ΣPCBs analysis due to limitations in available toxicity data for avian species for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (Giesy and Kannan, 1998; Giesy et al., 2000; Blankenship and Giesy, 2002; Giesy et al.., 2006).

Moreover, it has been suggested that TEQs based on PCBs may overestimate exposure relative to TCDD (Custer et al., 2005), so assessments based on TCDD exposure are likely to overestimate risk when applied to PCB exposure. Measures of condition of individual house wrens and eastern bluebirds (i.e., physical attributes, nest attentiveness, nestling growth) were not different between the Site and reference study areas. Similarly, most measures of reproductive success including, clutch size, hatching success, fledging success, and overall productivity were not different between the Site and reference study areas. Differences were observed for house wren fledging success, which was greater in the Site study area than the reference study areas, and eastern bluebird productivity, which was reduced in the Site study area relative to the reference study area. However, both of these parameters fall within normal ranges compared to other studies (Neigh et al., 2007).

2.4 Uncertainty

These conclusions are based on substantial Site-specific data, which was collected over multiple years to support multiple independent lines of evidence, all of which draw a similar conclusion. Thus, these results have a high degree of certainty relative to more generic assessments. However, some uncertainty associated with the assessments and conclusions remain, as discussed below.

2.4.1 Spatial applicability of terrestrial passerine data

Measured concentrations of ΣPCBs in passerine tissues, as well as supporting information on food web exposure, confirms the applicability of the selected study BSAs to other areas of floodplain soils. In order to provide insight into the relative exposures between the Site study area (defined as the former Trowbridge Impoundment) and other areas of floodplain soils along the river, adults and eggs of American robin were also collected from the former Plainwell Impoundment, which is a less expansive, but similarly contaminated area of floodplain soils approximately 20km upstream of the passerine study area but downstream of most point sources of ΣPCBs. Mean concentrations of ΣPCBs in adult American robins were ~60% less (0.7 mg ΣPCBs/kg ww, n=9) than those for robins from the Site study area (former Trowbridge Impoundment, 1.1 mg ΣPCBs/kg ww, n=8). Moreover, congener profiles and thus dioxin-like relative potency was similar between the two areas, indicating similar exposure pathways. This apparent similarity in exposure supports the use of Trowbridge as a conservative model for other impoundments along the River.

2.4.2 Representativeness of Selected Study Species

The house wren and eastern bluebird were selected to represent terrestrial insectivorous and omnivorous avian guilds respectively. As described previously, these two species were selected for a number of reasons including species characteristics, dietary composition, foraging characteristics, site use, exposure potential, and ease of study. To evaluate the representativeness of these two species for other omnivorous and insectivorous birds feeding in the floodplains – in particular the American robin, which was selected as the representative omnivorous bird in the Final Revised Baseline ERA (CDM 2003) – whole body ΣPCB tissue burdens were compared. The concentrations of ΣPCBs in adult American robins were substantially less than those of co-located house wrens (Table 2-3) indicating that robin exposure to ΣPCBs within the study area is less than other representative species.

While whole body eastern bluebird tissue data were not measured, the eastern bluebird is considered representative of other omnivorous birds, including the robin. Both in the family *Turdidae*, the American robin, and bluebird have similar dietary composition characteristics, feeding on a combination of plants and invertebrates. While their dietary composition is similar, the foraging patterns of the two species are slightly different, with the bluebird using a perch and pounce method and the American robin tending toward communal short grass foraging, especially early in the year during egg laying and brooding. Based on Site-specific observations of *Turdidae* foraging, it was noted that bluebirds tended to forage on-site almost exclusively, while American robins appeared to spend significant time off-site, above the floodplain in managed short grass habitat (e.g., lawns and golf courses proximal to the Site study area). Thus, based on Site-specific habitat characteristics and Site-specific observations, it was deemed that the American robin Site use was likely less than that of its bluebird cohort, thereby reducing its exposure potential to floodplain contaminants. Another factor that could contribute to the observed greater exposure (as measured by tissue concentrations) of bluebirds relative to robins is the finding that the Site-specific bluebird diet included some aquatic insects which may contain higher PCBs concentrations than the terrestrial species consumed by the robin (Kay et al., 2005).

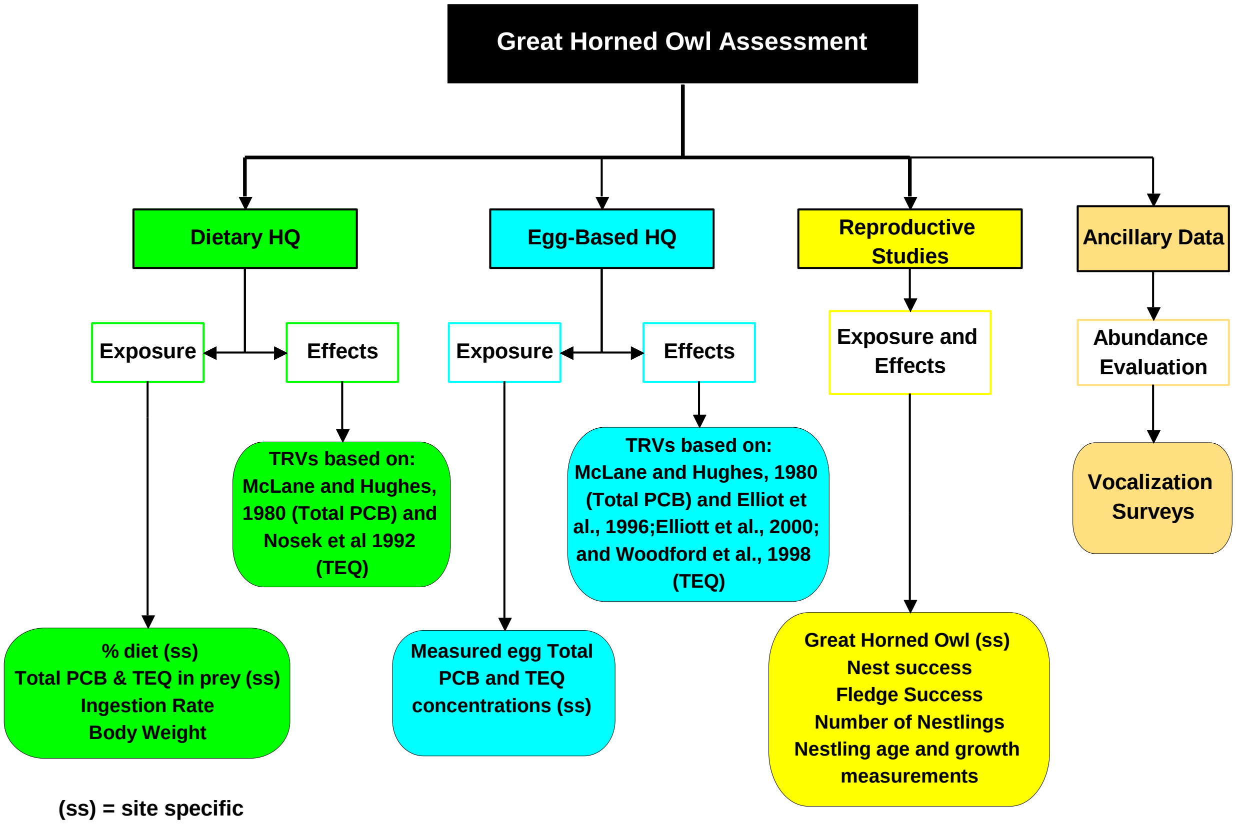
Given that these species are closely related, it was hypothesized that they would have similar sensitivity to adverse effects from the same toxicant. In fact, based on some recent work on the relative sensitivity among birds, it has been noted that there is a direct relationship between the specific amino acid sequence in the AhR ligand binding domain (AHRLBD), the binding affinity of dioxin like compounds ,and the sensitivity of the species to adverse effects (Karchner et al., 2006; Head et al., 2006). The AHRLBD for the American robin and the eastern bluebird are identical (Kennedy, Zwiernik, Giesy, unpublished data). Thus, based on an estimated greater exposure to ΣPCBs and similar sensitivity, individual and population measures were collected only for bluebirds and thought to be conservatively representative of other passerines, including the American robin.

3 Assessment of Potential Risk to Carnivorous Terrestrial Birds (Great Horned Owl)

The great horned owl (GHO) was the avian species evaluated in the Final Revised Baseline ERA (CDM, 2003) to represent the carnivorous feeding guild. Risks to the GHO were based on two lines of evidence: dietary exposures estimated through a food web model and comparisons of measured egg concentrations to tissue-based TRVs. The two approaches yielded conflicting results, with high risks predicted based on the egg concentration approach and comparatively low risks based on the food web model. In addition, CDM (2003) identified several uncertainties with the evaluated data. For example, there did not appear to be a clear link between the concentrations of PCBs in sediment and soil and the measured concentrations in egg tissues, particularly in the lower Kalamazoo River (i.e., downstream of Lake Allegan). With respect to the food web model, the dietary exposure estimate was based on a combination of measured (small mammal) and predicted (bird) prey tissue concentrations, however, Site-specific information about actual diet composition of GHO within the study area was very limited. CDM (2003) concluded that the GHO might be at significant risk depending on the actual composition of the diet but that because “the primary of route of exposure of great horned owls to PCBs is poorly understood at this site, protection of great horned owls and other similar birds should not be the basis of PRGs for floodplain sediment or surface soil.”

Based on the preliminary results presented by CDM (1996; 1999), we designed a multiple-lines-of-evidence approach to evaluate potential risks to the GHO from exposure to PCBs (Strause et al., 2007a, 2008; Zwiernik et al., 2007). Specifically, the assessment included three lines of evidence including a Site-specific study of productivity and two HQ assessments based on Site-specific exposure estimates. The productivity assessment measured reproductive performance at both the Site study area (Trowbridge) and reference site (Fort Custer). The HQ assessments included a dietary-based assessment relying on measured prey tissue concentrations and a tissue-based assessment focused on comparisons of measured egg concentrations to egg-based TRVs. Exposures and risks were calculated using both a ΣPCBs and TEQ approach (Zwiernik, et al., 2007; Strause, et al. 2007a; 2008). The lines of evidence and associated inputs evaluated in our assessment are summarized in Figure 3-1 (below).

Figure 3-1. Lines of Evidence Evaluated by MSU for the Great Horned Owl



As with other receptors, the GHO study focused primarily on the former Trowbridge Impoundment with Fort Custer as the reference study area (Strause et al., 2008). For the GHO, studies of active nests were expanded to additional locations including several marshes south of Lake Allegan referred to as the lower Kalamazoo River Study sites (LKRSS) (Strause et al., 2007a; Zwiernik, et al., 2007). However, the habitat in these marshes is quite different from that of the formerly impounded areas, which are the focus of the peer review. Given these habitat differences, it is unlikely that these data could be easily extrapolated to the formerly impounded areas. Therefore, we have not considered the data from the LKRSS in this summary.

3.1 Hazard Quotient Analyses

Risks to the GHO were also evaluated through two HQ assessments, one based on a dietary food web model to estimate a daily dose for comparison to a dietary TRV, and the other based on comparison of measured and estimated egg concentrations to tissue-based TRVs. Each of these elements and the resulting HQs for both ΣPCBs and TEQ are discussed in the following sections. This summary focuses on the Site-specific information only; information on the reference site (Fort Custer) can be found in Strause et al. (2007a; 2008) and Zwiernik et al. (2007).

3.1.1 Exposure Calculations

Exposure of GHO to ΣPCBs and TEQ was assessed in terms of daily dose for dietary exposure and direct measures of tissue concentrations in nestlings and eggs. Because TRVs are not available for nestling tissues, the exposure estimates described below focus on dietary and egg-based exposures. In addition, this summary focuses only on dose estimates derived from Site-specific data such as measured concentrations of ΣPCBs and TEQ in eggs and Site-specific estimates of diet. The related publications (Strause, et al., 2007a, b, and 2008; Zwiernik, et al., 2007) also include discussions of egg concentrations predicted from the measured nestling blood plasma concentrations as well as dose calculations based on dietary compositions reported in the literature.

3.1.1.1 Dietary Exposure Estimate

Dietary exposure for GHOs was estimated by calculating APDD for ΣPCBs and TEQ. The dose model used is consistent with the model recommended in the USEPA's Wildlife Exposure Factors Handbook (USEPA, 1993) and is described in more detail in Strause et al. (2008). Site-specific inputs to the APDD include Site-specific measures of dietary composition and measured prey tissue concentrations. One of the key uncertainties noted in the Final Revised Baseline ERA (CDM, 2003) was lack of information about the GHO's diet, therefore, relative proportions of prey items in the Site-specific diet were determined by examining unconsumed prey remains as well as the skeletal remains in regurgitated pellets (Strause et al., 2008; Zwiernik et al., 2007). Based on this analysis, as well as information reported in the literature for a separate population of Michigan GHO (Craighead and Craighead, 1956), six general categories of prey were identified: passerine birds, waterfowl, mice/voles, shrews, muskrats, and rabbit/squirrel. Site-specific dietary composition results are summarized in Table 3-1 . Prey species identified in diet were subsequently collected for the purpose of developing Site-specific tissue concentrations. Specifically, whole-body prey item samples were collected from the Site study area as described in Strause et al., (2008) and Zwiernik et al. (2007) and evaluated for ΣPCBs, TEQ and lipid content. These data were combined with data previously collected, including 17 waterfowl samples as well as data for chipmunk and red squirrel to develop geometric mean PCBs and TEQ concentrations (U95 CL) for each species represented in the diet of the GHO (Table 3-2).

Table 3-1. Great Horned Owl Site-Specific Spring Diet Composition

Prey Type	Numeric Basis (percent occurrence)	Mass Basis (percent contribution)
Class Aves	**27.5**	**24.8**
Passerine[1]	25.8	22.0
Waterfowl[2]	1.7	2.8
Class Mammalia	**72.5**	**75.2**
Mice/Vole	49.0	6.0
Shrew	2.0	0.2
Muskrat	5.5	19.0
Rabbit[3]	16.0	50.0

Notes:
1. Passerine category includes all terrestrial birds and all unidentified bird ("unknown bird") remains.
2. Waterfowl category includes all aquatic birds.
3. Rabbit category includes squirrels and all unidentified medium-size mammal ("unknown mammal") remains.

Table 3-2. Geometric Mean and U95 CL Concentrations in Prey Items Collected from the former Trowbridge Impoundment

	Trowbridge				
	Number of Samples	**Range (ug/g ww)**	**Geometric Mean (ug/g ww)**	**U95 CL (ug/g ww)**	**Lipid (%)**
	ΣPCBs				
Passerines[1]	20	0.06 – 32	1.3[2]	3.1	4.90
Waterfowl[3]	17	0.13 – 28	0.89	1.8	3.80
Mice/Vole	20	0.03 – 0.6	0.07[2]	0.10	4.57
Shrew	17	0.03 – 3.2	0.85[2]	1.5	2.68
Muskrat	7	0.01 – 0.11	0.06[2]	0.09	2.07
Rabbit[4]	1	0.57	0.57	0.57	4.95
Crayfish	13	0.08 – 1.9	0.37[2]	0.60	1.62
	TEQ				
Passerines[1]	19	0.001 – 4.8	0.06[2]	0.19	0.005
Waterfowl[3]	17	0.04 – 7.7	0.24	0.72	0.004
Mice/Vole	20	0.00047 – 0.004	0.0008	0.001	0.005
Shrew	17	0.004 – 0.25	0.05[2]	0.08	0.003
Muskrat	7	0.0043 – 0.05	0.01	0.05	0.002
Rabbit[4]	1	0.07	0.07	0.07	0.005
Crayfish	13	0.003 – 0.37	0.06[2]	0.11	0.002

Notes:
1. Geometric mean total PCBs concentrations in Trowbridge starling (1), house wren (6), tree swallow (5), and American robin (8 for total PCB, 7 for TEQ) used as representative of terrestrial passerine concentrations.
2. Trowbridge concentrations were significantly greater than those at Fort Custer, the reference site ($p \leq 0.01$)
3. Waterfowl samples were collected from 5 locations on the Kalamazoo River and were not divided between upstream and downstream sampling locations because of uncertain local residence status on the river.
4. Total PCBs and TEQ concentrations in Trowbridge chipmunk (1) used as surrogate value for rabbit concentration.

Using the Site-specific prey tissue concentrations and dietary composition information, the APDD for ΣPCBs and TEQ were calculated for GHOs (Strause et al., 2008; Zwiernik et al., 2007). It was assumed that GHO at the study area will obtain 100% of their diet requirements from the 100-yr floodplain (Site use factor = 1). In addition, incidental soil ingestion was calculated for the Site study area GHOs as described by Strause et al. (2008). A summary of the APPD developed is presented in Table 3-3.

Table 3-3. Range of Average Potential Daily Doses[1]

	Trowbridge PCB-Based Dietary Models	
	ΣPCB	TEQ
Site-Specific APDD (ug/g-day)		
Geometric Mean	0.03 – 0.04	0.002 – 0.004
U95 CL	0.059 – 0.061	0.005 – 0.007

Notes:
1. The range of calculated Average Potential Daily Doses (APDD) results from using site-specific diet estimations based on both total frequency (numeric-basis) and biomass contribution (mass-basis). Includes incidental ingestion of floodplain soils at the former Trowbridge impoundment.

3.1.1.2 Tissue Based Exposure Estimates

Exposure of GHO to ΣPCBs and TEQ were also evaluated by collecting and measuring ΣPCBs and TEQ concentrations in eggs from nests within the reference site (Fort Custer) and the Site study area (former Trowbridge Impoundment) (Strause et al., 2007a, b; Zwiernik et al., 2007). Nineteen eggs were collected from the Site study area and evaluated for PCBs. Fresh eggs were collected as soon as possible following confirmed initiation of incubation while addled eggs were collected four to six weeks post hatch or in instances when nest abandonment had occurred. A summary of the measured egg concentrations is provided in Table 3-4 (Strause et al., 2007a).

Table 3-4. Measured Egg Concentrations in Resident Great Horned Owls

	n	Range	Geometric Mean
ΣPCB (ug/g)	6	0.53 – 4.4	1.4
TEQ (ug/g)	5	0.000007 – 0.00003	0.000014

3.1.2 Effects Levels (i.e., Toxicity Reference Values)

TRVs were derived from information in the literature according to USEPA guidance for both dietary and egg-based exposures. The selection of TRVs for the GHO was based on the same criteria described in Section 2.1 for the passerine species but using data from controlled dose-response studies of screech owl (*Otus asio*) (McLane and Hughes, 1980).

3.1.2.1 Dietary TRVs

TRVs for ΣPCBs used in the dietary exposure assessment were based on values reported in the literature for NOAELs and LOAELs for ΣPCBs. The dietary NOAEL of 0.4 ug ΣPCBs/g/day for GHO was based on a controlled, laboratory study on the reproductive effects of ΣPCBs on the screech owl (McLane and Hughes, 1980). A LOAEL was not identified in this study. Therefore, a TRV was estimated by applying a NOAEL to LOAEL uncertainty factor of 3, resulting in a LOAEL TRV of 1.2 ug ΣPCBs/g/day.

No studies of the effects of TEQ were available for deriving TRVs, and no studies were found in which there was a closely related test species to GHO. In the absence of such data, a dietary TEQ-based LOAEL of 0.0014 ug TEQ/g bw/d for GHO was derived (Strause et al., 2008) based on a sub-chronic laboratory study evaluating the effects of intra-peritoneal injections of 2,3,7,8-TCDD at concentrations of 0.0001 ug TCDD/g/week on ring-necked pheasants (Nosek et al., 1992). A NOAEL (0.000014 ug TEQ/g bw/d) was derived as well by applying a safety factor of 10 (Strause et al., 2008).

3.1.2.2 Egg-based TRVs

TRVs were used to evaluate the potential for adverse effects due to concentrations of ΣPCBs in GHO eggs. The TRV values selected were based on a feeding study with screech owls (McLane and Hughes, 1980). The NOAEL in this study was 7 ug ΣPCBs/g/day. A LOAEL was not determined in this study. Therefore, a TRV was estimated by multiplying the NOAEL value by an uncertainty factor of 3, resulting in a LOAEL TRV of 21 ug ΣPCBs/g/day.

No relevant studies on effects of TEQ in the eggs of owl species were available from which to derive a TRV. Therefore, a tissue-based NOAEC for TEQ in GHO eggs of greater than 0.001 ug TEQ/g egg (wet wt) was estimated (Strause et al., 2007a) based on the results of studies evaluating bald eagle chicks (presented on an egg basis (Elliot et al., 1996) and osprey egg exposures (Elliott et al., 2000; Woodford et al., 1998). A LOAEC concentration of 0.004 ug TEQ/g egg (wet wt) was applied, based on CYP1A induction in bald eagles (Strause et al., 2007a).

3.1.3 Results

Both dietary and egg-based HQs for ΣPCBs and TEQ are described below.

3.1.3.1 Dietary HQs

Dietary HQs were calculated based on the range of APDD values encompassing the geometric mean and associated U 95CL values for each respective prey item and the dietary proportion contributed by each prey item compiled on both a numeric- and mass-basis (Strause et al., 2008). All HQs determined for geometric mean and U 95CL exposures were less than 1.0 (Table 3-5, below).

Table 3-5. Site-Specific Dietary Hazard Quotient values for Great Horned Owls[1]

	HQ NOAEL	HQ LOAEL	HQ NOAEL	HQ LOAEL
	ΣPCBs		TEQ	
Geometric Mean	0.07 – 0.09[2]	0.02 – 0.03	0.17 – 0.27	0.02 – 0.03
U95 CL	0.14 – 0.15	0.05	0.38 – 0.50	0.04 – 0.05

Notes:
1. HQs calculated based on toxicity reference values provided in Strause et al. (2007a) and APDD based on results of field collected great horned owl pellets and prey remains from active nests at each Kalamazoo River study site.
2. The range of calculated Average Potential Daily Doses (APDD) results from using diet estimations based on both total frequency (numeric-basis) and biomass contribution (mass-basis) (see Strause et al., 2007a, Table 2).

3.1.3.2 Egg Based HQs

Egg-based HQs were calculated by dividing concentrations of ΣPCBs and TEQ in GHO eggs by the egg-based NOAEL and LOAEL TRVs. The measured geometric mean and U 95CL concentrations of ΣPCBs TEQ in the eggs from the Site study area were all below the egg-based NOAEL TRVs (greatest NOAEL based HQ = 0.05), therefore, no adverse effects were predicted by this assessment (Strause et al., 2007a).

3.2 Measures of Reproductive Success

The Site-specific GHO population in the Upper Kalamazoo River was assessed through an evaluation of reproductive performance of GHO populations in both the Site study area (former Trowbridge Impoundment) and reference site (Fort Custer) (Strause et al., 2007a, 2008; Zwiernik et al., 2007). Information collected at each of seven active nests (one in Fort Custer, six in Trowbridge) included: nest success, number of nestlings per nest, fledgling success, and nestling age and growth measurements.

Abundance of GHO was also estimated using a vocalization survey in which GHO hoots were broadcast to provoke responses from both adult and juvenile GHO. A total of 46 successful call-response surveys were conducted from 2000 to 2002. A more detailed discussion of these methods is provided in Zwiernik et al., (2007) and Strause et al. (2007a, 2008).

Over the three-year study period from 2000 to 2002, productivity (fledglings/active nest) at active nests was monitored. Reproductive success was similar at both study areas, averaging 1.0 successful fledge per active nest at each location (Table 3-6, below; Strause et al., 2007a; 2008). These results are consistent with productivity measurements for natural Midwestern GHO populations residing in varied upland habitats (Holt, 1996) suggesting that despite the difference in PCBs exposure at the Site study area relative to the reference site, the reproductive success of the population does not appear to be impacted (Strause et al., 2007a; 2008). Similarly, Site-specific measures of abundance and use indicate the Site study area (the former Trowbridge Impoundment) GHO populations were near the expected carrying capacity (approximately one pair per 1600 hectares (ha) and a total of three pairs in the Trowbridge floodplain per Houston et al. [1998]). Furthermore, nest acceptance rates and nest fidelity of actively breeding GHOs across all nesting seasons included in the study were consistent with previous studies of artificial nest acceptance and habitat usage by *Strigiforms* in Midwestern forests (Strause et al., 2007a; 2008).

Table 3-6. Relative Abundance and Productivity of Resident Great Horned Owls at Ft. Custer (Reference) and the Former Trowbridge Impoundment

Study Year	**2000**		**2001**		**2002**		**All Years (2000 – 2002)**	
	Ft. Custer	**Trowbridge**	**Ft. Custer**	**Trowbridge**	**Ft. Custer**	**Trowbridge**	**Ft. Custer**	**Trowbridge**
Relative Abundance[1]	**N2=4**	**N2=7**	**N2=9**	**N2=7**	**N2=11**	**N2=8**	**N2=24**	**N2=22**
Adults	Mean Response Rate[3]		Mean Response Rate[3]		Mean Response Rate[3]		Mean Response Rate[3]	
Total[4]	2.5	2.57	0.89	2.71	0.55	3	1.31	2.76
Foraging[5]	1.5	1.43	0.67	1.86	0.37	1.63	0.85	1.64
Paired[6]	1	1.14	0.22	0.86	0.18	1.38	0.47	1.13
Juveniles	Response Frequency[7] n, (%)		Response Frequency[7] n, (%)		Response Frequency[7] n, (%)		Response Frequency[7] n, (%)	
Fledgling[8]	0(0)	0(0)	1(11)	7(100)	0(0)	1(12)	1(04)	8(36)
Productivity9	N^{10}=0	N^{10}=1	N^{10}=1	N^{10}=2	N^{10}=0	N^{10}=3	N^{10}=1	N^{10}=6
Fledglings	0	1	1	4	0	1	1	6
Fledglings/ Nest	0	1	1	2	0	0.3	1	1

Notes:

1. Relative abundance estimates derived from hoot call/response surveys completed at dawn and dusk.
2. N=number of complete surveys.
3. Mean response rate is averaged across N completed surveys for each year. All years mean is the mean of the means.
4. Includes discrete responses from both individual and paired (male + female) owls.
5. Includes responses from unpaired individuals only.
6. Includes responses from paired (male + female) owls only.
7. Average response frequency of fledgling owls (n=number (percent) of completed surveys with at least one fledgling begging call response) expressed on a yearly basis, and averaged over all years.
8. A measure of current and successful breeding activity.
9. The total number of successful fledglings from all active nests (# fledglings/# active nests) per year within each sampling area, and averaged over all years/sum total of all active nests.
10. N=number of active nests.

3.3 Summary and Discussion

This assessment employed multiple lines of evidence to minimize uncertainty in assessment endpoints and to provide a rigorous basis for remedial decision-making. Results of both the dietary and tissue based hazard assessments suggest that GHO populations residing in the reference and Site study areas are not at risk for effects induced by PCBs or $TEQ_{WHO\text{-}Avian}$ in floodplain soil, (Strause et al., 2007a; 2008; Zwiernik et al., 2007). This conclusion is consistent with measurements of fledgling productivity, which indicated that reproductive success in the Site-specific population was consistent with that expected in healthy, Midwestern populations (Strause et al., 2007a). The results of these studies suggest that current concentrations of ΣPCBs and $TEQ_{WHO\text{-}Avian}$ in the former impoundments are not sufficient to pose a significant risk to GHO populations.

3.4 Uncertainty

As described for the passerines, the conclusions for the GHO are based on substantial Site-specific data, collected over multiple years to support multiple independent lines of evidence, all of which draw a similar conclusion. Thus, these results have a high degree of certainty relative to more generic assessments. However, some uncertainty associated with the assessments and conclusions remain and they are discussed below.

3.4.1 Dietary Exposures

This assessment relied on an analysis of Site-specific pellet and prey remains to provide an estimate of the dietary composition of the GHO residing along the Kalamazoo River. There are uncertainties associated with these methods. For example, it is possible to overestimate the frequency of occurrence of large prey items because of their tendency to be represented in more than one pellet or prey remain sample. To reduce this uncertainty, all pellet and prey remains were combined into two separate composite samples for each field season, one approximately 4-6 weeks post hatch and one post-fledge. Conversely, very small prey items may be under represented because they may be completely digested. However, small prey items are unlikely, on a mass basis, to comprise a significant proportion of the overall diet for most individuals. Additionally, many small prey items are likely to be very young animals, which are unlikely to have accumulated significant concentrations of PCBs prior to being consumed

GHO diets have been shown to vary by season, habitat, and prey availability. These variables were controlled by targeting riparian floodplain habitats that were buffered from most human disturbances and within 100 meters (m) of the river to provide uniformity in foraging habitat and available prey populations. In addition, pellets and prey remains were collected from multiple years and always during active nesting and brooding periods. Results for the Site and reference study areas did exhibit differences in the proportion of passerine versus terrestrial birds and in the proportion of rabbits (Strause et al., 2008). However, when considered on a class rather than species basis, these differences were less significant.

The daily dose estimates were based on concentrations measured in prey species collected from the Site study area. To the extent possible, prey species evaluated were based on the Site-specific dietary composition determined for the study area, however, there were a few species that could not be collected (i.e., rabbit, pheasant and grey/fox squirrel) and data for surrogate species were used. All surrogate data used to address these data gaps were selected in a conservative manner, so it is likely that the estimated dose is greater than the actual dose to GHO at the study area.

3.4.2 Toxicity Reference Values (TRVs)

There were no available toxicity studies evaluating the effects of TEQ on GHO. For the egg-based TRV, studies focusing on the effects of TEQ on eagles and ospreys were used to derive estimated TRVs. Although these data are not specific to GHO, these species are carnivorous and, therefore, represent an appropriate trophic level. In addition, at the concentrations selected there were no adverse effects on developmental or any other ecologically relevant endpoint. Therefore, these concentrations are expected to be appropriately conservative to be protective of the GHO.

For the dietary exposures, a study evaluating the effects of intra-peritoneal injections of 2,3,7,8-TCDD in ring-necked pheasants was used as the basis for the TRV. Use of a TRV based on 2,3,7,8-TCDD exposure is likely to overestimate risks when applied to PCB exposure. In addition, tolerance TEQs exposure by birds is species specific and available information indicates that raptors (e.g., American kestrel, osprey, and bald eagle) are less sensitive than *Galliformes* (e.g., ring-necked pheasants). Because GHOs are more similar to raptors, it is assumed that they would also be more tolerant than ring-necked pheasants. Based on these considerations, these TRVs were determined to be appropriately conservative to be protective of the GHO.

3.4.3 Productivity Assessment

The assessment of reproductive health is based on a relatively small sample size (i.e., seven active nests). In addition, the results for the reference area were confounded by elevated adult mortality. However, the measures of fledgling success and species abundance were consistent with those reported in the literature for other healthy GHO populations.

4 Assessment of Potential Risk to Small Terrestrial Mammals (Short-Tailed Shrew)

In the CDM Final Revised Baseline ERA (2003), the white-footed mice and deer mice were identified as the representative receptors for evaluation of small burrowing terrestrial mammalian species. These species represent an omnivorous feeding guild. The CDM Final Revised Baseline ERA found that 1) omnivorous terrestrial species, represented by mice, are unlikely to be at risk unless they reside in the most contaminated areas and 2) ΣPCBs accumulation by mice appears small. These conclusions were based primarily on a dietary HQ analysis and measured concentrations of ΣPCBs in mice tissues (CDM, 2003) and were not available at the time that our studies were undertaken. Therefore, we completed an evaluation of risk to shrews, a carnivorous burrowing small mammal with high potential for exposure to PCBs in soils of the formerly impounded areas.

The short-tailed shrew (*Blarina brevicauda*) lives on and in the soil and eats soil invertebrates, including insect larvae of the orders *Diptera*, *Lepidoptera*, *Orthoptera*, and *Hymenoptera* and adults of the order *Coleoptera*. It also eats beechnuts (Hamilton, 1930; 1941), earthworms (Babcock, 1914; Whitaker and Ferraro, 1963), and snails (Shull, 1907). Because of its small size, the short-tailed shrew has the greatest weight-specific rate of metabolism among mammals (Lawlor, 1979). Given its carnivorous diet consisting largely of soil invertebrates, its high ingestion rate, and relatively small home range (varies from 0.39 ha to 0.96 ha; Buckner, 1966; Faust et al., 1971), the shrew is likely to be one of the most exposed terrestrial species in the areas of the former impoundments. Therefore, the shrew was selected as a surrogate species for small burrowing terrestrial mammalian species. The shrew evaluation included here consists primarily of a dietary HQ analysis as described below.

4.1 Dietary Exposure Calculations

Risk to shrews is based on measured Site-specific dietary exposure to ΣPCBs and a literature-derived TRV. The dietary exposure for resident shrews was calculated based on EPA guidelines in the Wildlife Exposure Factors Handbook (USEPA, 1993). The dietary composition for resident shrews was based on literature values and confirmed by examining the stomach content of trapped shrews. The identified dietary items were then sampled from each of the same locations that the shrews were collected at three time points during the spring and summer and analyzed for ΣPCBs. This allowed for a more Site-specific assessment of the APDD to estimate exposure. The APDD for the shrew based on mean dietary concentrations was 0.18 mg ΣPCBs/kg/day and based on the U95 CL was 0.33 mg ΣPCBs/kg/day.

4.2 Dietary Toxicity Reference Value

The dietary TRV was derived from a two-generation reproduction study in which Sherman-strain rats were exposed by diet to five doses of Aroclor 1254 (0, 1, 5, 20, or 100 mg/kg in feed) for up to 274 days (d) through a critical reproductive life stage (Linder et al., 1974). No adverse effects were observed at a dose level of 5 mg/kg in feed which represents a dose of 0.32 mg/kg/d. Furthermore, since Aroclor 1254 (from the laboratory study) is more toxic than Aroclor 1242 (the predominant Aroclor found in the Kalamazoo River AOC), the TRV is based on the more toxic of the potential Aroclors (Harris et al., 1993). Thus, 5 mg/kg, bodyweight (bw) (or 0.32 mg Aroclor 1254/kg bw/d) in feed should be considered a conservative estimate of the NOAEL. At 20 mg, Aroclor 1254/kg, ww in feed (or 1.5 mg Aroclor 1254/kg/d), adverse effects were observed including a reduction in litter size. Since the study considered dietary exposure during the sensitive and ecologically relevant time period of reproduction, and was extended over two generations, the 0.32, and 1.5 mg Aroclor 1254/kg bw/d doses were considered to be chronic dietary-based NOAELs and LOAEL values, respectively.

In extrapolating from toxicity data on rats to shrews it is important to account for physiological factors such as metabolic rates, and responses to toxic chemicals, which are functions of body size. Smaller animals have greater metabolic rates and are usually more resistant to toxic chemicals because of more rapid rates of detoxification or elimination (Sample et al., 1996). To account for these size differences, NOAEL and LOAEL TRVs from the test species (rats) were adjusted for differences in body size as compared to the wildlife species (shrews) using methodology from Sample et al., (1996) and USEPA Guidance (USEPA, 1992). Using the mean body weight of controls, rats in the Linder et al., (1974) study (mean = 425 g) and the mean body weight of control shrews in the Russell (1998) study (mean = 19.4 g), the adjusted dietary-based TRVs for shrews are 0.69 and 3.2 mg ΣPCBs /kg bw/d for the NOAEL and LOAEL, respectively (Equation 4-1).

$$NOAEL_{wildlife} = NOAEL_{test\ species} * [\text{body weight}_{test\ species}/\text{body weight}_{wildlife}]^{(1-0.75)}$$ **[Equation 4-1]**

4.3 Hazard Assessment

Hazard quotients were calculated for short-tailed shrews based on several assumptions regarding dietary exposure (Table 4-1). The HQs based upon the geometric mean of the range of Site-specific exposures are 0.06 and 0.26 for LOAEL and NOAEL based TRVs, respectively. The HQs based upon the U95 CL of geometric mean of Site-specific exposure are 0.10 and 0.48 for LOAEL and NOAEL based TRVs, respectively.

Table 4-1. Hazard Quotients for Shrews Calculated Based on Predicted Dietary Exposure

Exposure Level	Dietary-Based Hazard Quotient	
	NOAEL-Based	LOAEL-Based
Geometric mean	0.26	0.06
U95 UCL geometric mean	0.48	0.10

4.4 Conclusions

The dietary HQs are based on Site-specific information on prey items in the shrew diet and on co-located data on PCBs concentrations in those prey items. All of the HQs, regardless of assumptions, are less than 1.0. Moreover, the HQs for the carnivorous shrew are consistent with those for the white-footed/deer mouse of 0.2 and 0.7 for the LOAEL and NOAEL as presented in the Final Revised Baseline ERA (Table 5-3 in CDM, 2003). Together these data do not indicate evidence of risk to either carnivorous or omnivorous small mammals in floodplains of the formerly impounded areas.

4.5 Uncertainty

The literature-based TRV represents the primary source of uncertainty in this evaluation of risk to shrews in the floodplains of the former impoundments on the Kalamazoo River. This study evaluates risk to rats and requires extrapolation to shrews based on differences in body weight. However, this study is of high quality. It evaluates reproduction, an ecologically important endpoint that is sensitive to ΣPCBs, over two generations and exposure is based on Aroclor 1254 which is more toxic than Aroclor 1242, the primary form of PCBs in paper waste generated by the KRSG facilities. Moreover, the extrapolation from data on rats to shrews made use of mechanistically-based USEPA methods. This extrapolation is therefore unlikely to underestimate risk to small mammals in floodplains of the former impoundments.

Trapping studies conducted in the Site study area at the former Trowbridge Impoundment and reference study area at Fort Custer support the conclusions from the HQ analysis. The trapping success for small mammals was greater at the former Trowbridge Impoundment locations (0.0446 small mammals per trap-night) compared to the Fort Custer locations (0.0125 small mammals per trap night) (Table 4-2). Trapping success for shrews was also somewhat greater at the Trowbridge locations (0.007 shrews per trap night) than at the reference locations (0.004 shrews per trap night). These data indicate that small mammals including shrews are at least as abundant in the former Trowbridge Impoundment as in the reference area at Fort Custer.

Table 4-2. Results of Small Mammal Trapping at the Former Trowbridge Impoundment and at Fort Custer (Reference) Locations Collected During the 2000 Summer Season

Location	# Trap Nights[1]	Shrews[2]	Meadow Voles	Deer Mice	White-footed Mice	Old field Mice	Jumping Mice	Chipmunks and Squirrels	Total # of Small Mammals
Trowbridge 1	784	4	16	6			1		
Trowbridge 2	392	3	6	17					
Trowbridge 3	588	4	14	6			1		
Trowbridge 4	588	6	11	7	1		1	1	
All Trowbridge	**2352**	**17**	**47**	**36**	**1**	**0**	**3**	**1**	**105**
Fort Custer 1	2610	6		12		1	1	4	
Fort Custer 2	1305	10		3	3		4	5	
All Fort Custer	**3915**	**16**	**0**	**15**	**3**	**1**	**5**	**9**	**49**

Notes:
1. Number of trap nights was calculated as the number of traps set out multiplied times the number of nights that they were in place.
2. Shrews include both the short-tailed shrew and the masked shrew.

5 Summary

We performed a series of comprehensive, Site-specific, congener-specific, multi-year, multiple lines of evidence studies to evaluate potential risk to representative ecological receptors posed by elevated concentrations of PCBs in floodplain soils in the formerly impounded areas of the Kalamazoo River. Terrestrial receptor groups evaluated included omnivorous (eastern bluebird), insectivorous (house wren), and carnivorous (great horned owl) birds, as well as small carnivorous mammals (short tailed shrew). The information was collected with the primary goal of reducing uncertainty in the Baseline ERA (CDM 2003) by minimizing reliance on corollary assumptions.

Avian studies included productivity measures for the eastern bluebird, house wren, and GHO, as well as measures of Site-specific dietary and tissue-based exposure. Most measures of reproductive success including, clutch size, hatching success, fledging success, and overall productivity were not different between the Site and reference study areas for GHO, house wrens and bluebirds. Differences were observed for house wren fledging success, which was greater in the Site study area than the reference study areas, and eastern bluebird productivity, which was reduced in the Site study area relative to the reference study area. However, both of these parameters fall within normal ranges compared to other studies (Neigh et al., 2007). For owls, reproductive success in the Site-specific population was consistent with that expected in healthy, Midwestern populations (Strause et al., 2007a). The HQ analyses indicated that terrestrial passerines in the former Trowbridge Impoundment are accumulating increased concentrations of ΣPCBs relative to the reference study area (Neigh et al., 2006b), but multiple lines of evidence, taken together, indicate that this increased exposure to ΣPCBs in floodplain soils is not associated with increased risk to these species. The HQ analysis for the owl indicated that GHO populations residing in the reference and Site study areas are not at risk for effects induced by PCBs (either as SPCB or TEQ) in floodplain soil, (Strause et al., 2007a; 2008; Zwiernik et al., 2007).

Mammalian studies primarily included collection of site specific tissue data to support calculation of site-specific exposure estimates for the short tailed shrew. All of the dietary HQs were less than 1.0 indicating little to no potential risk to these receptors. Moreover, the HQs for the carnivorous shrew are consistent with those for the white-footed/deer mouse presented in the Final Revised Baseline ERA (Table 5-3 in CDM, 2003). Together these data do not indicate risk to either carnivorous or omnivorous small mammals in floodplains of the formerly impounded areas.

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